PB89-198006

Experimental Acidification of Little Rock Lake (Wisconsin) Fish Research Approach and Early Responses

(U.S.) Environmental Research Lab.-Duluth, MN

1989

U.S. Department of Commerce
National Technical Information Service
NTIS

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO.	2.	3.
EPA/600/J-89/023		PB89-198006

4. TITLE AND SUBTITLE	5. REPORT DATE
Experimental Acidification of Little Rock Lake (Wisconsin): Fish Research Approach and Early Responses.	1989
	6. PERFORMING ORGANIZATION CODE

7. AUTHOR(S)	8. PERFORMING ORGANIZATION REPORT NO.
W.A. Swenson, J. H. McCormick, T.D. Simonson, K.M. Jensen and J.G. Eaton	

9. PERFORMING ORGANIZATION NAME AND ADDRESS	10. PROGRAM ELEMENT NO.
U.S. Environmental Protection Agency Environmental Research Laboratory-Duluth 6201 Congdon Boulevard Duluth, MN 55804	11. CONTRACT/GRANT NO.

12. SPONSORING AGENCY NAME AND ADDRESS	13. TYPE OF REPORT AND PERIOD COVERED
Environmental Research Laboratory Office of Research and Development U.S. Environmental Protection Agency Duluth, MN 55804	14. SPONSORING AGENCY CODE EPA-600/03

15. SUPPLEMENTARY NOTES

Arch. Environ. Contam. Toxicol. 18:167-174, January 1989.

16. ABSTRACT

One goal of research at Little Rock Lake, Wisconsin, is to enhance understanding of lake acidification effects on warm- and cool-water fishery resources. The Little Rock Lake fish assemblage is characteristic of many acid sensitive waters in North America and is dominated by yellow perch (Percidae) and sunfishes (Centrarchidae). Analyses of reproduction, early survival and growth rates in the field were designed around the differing reproductive modes of these taxa. Complementary laboratory research on early life stages was conducted to assist in isolating direct effect mechanisms and to determine the reliability of laboratory results in predicting field response.

Preliminary findings suggest that lake acidification to pH 5.6 has not influenced reproductive activity of the four most abundant fish species. However, the field results suggest that year-class failure of rock bass (*Ambloplites rupestris*) may be occurring due to reduced survival of early life stages. Reduced growth and food conversion efficiency of Age 0 largemouth bass (*Micropterus salmoides*) is also suggested. The laboratory bioassays indicate rock bass is the most acid-sensitive Little Rock Lake species tested. However, rock bass fry survival was not significantly affected until pH was reduced from 5.6 to 5.0.

(Copyright (c) 1989 Springer-Verlag)

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group

REPRODUCED BY
U.S. DEPARTMENT OF COMMERCE
NATIONAL TECHNICAL INFORMATION SERVICE
SPRINGFIELD, VA. 22161

18. DISTRIBUTION STATEMENT	19. SECURITY CLASS (This Report)	21. NO. OF PAGES
Release to public	Unclassified	11
	20. SECURITY CLASS (This page)	22. PRICE
	Unclassified	

NOTICE

This document has been reviewed in accordance with U.S. Environmental Protection Agency policy and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

This article documents research that was conducted under the auspices of the U.S. Environmental Protection Agency (USEPA). As an arm of the government, the USEPA has a paid up, non-exclusive, irrevocable, worldwide license to reproduce, prepare derivative works, and distribute to the public all of the material contained herein, and to grant others acting on its behalf the right to do so. The USEPA grants to the National Technical Information Service, Department of Commerce, the right to act as its agent in the distribution of this material.

EPA/600/J-89/023

Arch. Environ. Contam. Toxicol. 18, 167–174 (1989)

NTIS is authorized to reproduce and sell this report. Permission for further reproduction must be obtained from the copyright owner.

Archives of Environmental Contamination and Toxicology

© 1989 Springer-Verlag New York Inc.

Experimental Acidification of Little Rock Lake (Wisconsin): Fish Research Approach and Early Responses

William A. Swenson*,**, J. Howard McCormick***, Timothy D. Simonson**, Kathleen M. Jensen†, and John G. Eaton***

*Division of Science and Mathematics, **Center for Lake Superior Environmental Studies, University of Wisconsin, Superior, Wisconsin 54880, ***U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, Minnesota 55804, and †American Scientific International Inc., McLean, Virginia 22101

Abstract. One goal of research at Little Rock Lake, Wisconsin, is to enhance understanding of lake acidification effects on warm- and cool-water fishery resources. The Little Rock Lake fish assemblage is characteristic of many acid sensitive waters in North America and is dominated by yellow perch (Percidae) and sunfishes (Centrarchidae). Analyses of reproduction, early survival and growth rates in the field were designed around the differing reproductive modes of these taxa. Complementary laboratory research on early life stages was conducted to assist in isolating direct effect mechanisms and to determine the reliability of laboratory results in predicting field response.

Preliminary findings suggest that lake acidification to pH 5.6 has not influenced reproductive activity of the four most abundant fish species. However, the field results suggest that year-class failure of rock bass (*Ambloplites rupestris*) may be occurring due to reduced survival of early life stages. Reduced growth and food conversion efficiency of Age 0 largemouth bass (*Micropterus salmoides*) is also suggested. The laboratory bioassays indicate rock bass is the most acid-sensitive Little Rock Lake species tested. However, rock bass fry survival was not significantly affected until pH was reduced from 5.6 to 5.0.

The rationale for the experimental ecosystem manipulation approach and the general research design of the Little Rock Lake (LRL), Wisconsin acidification project are described by Watras and Frost (1989) in this issue. Briefly, the design consists of separating the two similar basins of this low alkalinity (≅25 μg/L), 18 ha lake with a plastic seacurtain and gradually acidifying one basin. Sulfuric acid is being added to reduce the pH by about 0.5 units every other year. After 1½ years of baseline study (1983–1984), the treatment basin pH was reduced from 6.1 to 5.6 during 1985–1986 and to 5.1 during 1987 (Watras and Frost 1989). After another year at pH 5.1, two years at pH 4.5 will follow. The acidification is accompanied by detailed studies of the important chemical and biological components in the system (Watras and Frost 1989).

This paper describes field and laboratory research conducted on the fish species of Little Rock Lake. While the focus is on experimental design, results have been included on the pH 5.6 field experiment and laboratory experiments with early life stages of three major fish species. The experimental design for the fish research was developed around the major LRL project goals (Watras and Frost 1988) and several fish project objectives. All results from the field experiment are considered to be preliminary.

One primary goal of LRL acidification is to expand insights gained through the experimental acidification of the coldwater system, Lake 223, Ontario, (Schindler 1987; Mills *et al.* 1987), to warm- and cool-water systems. An important consideration in selecting LRL was that the fish assemblage represented many acid-sensitive lakes in the northeastern, southeastern and central U.S. and in central and eastern Canada. The community includes 11 fish species, four of which are abundant enough for tracking population-level changes.

Yellow perch (*Perca flavescens*), the most abundant species, is considered quite resistant to acidification (Magnuson *et al.* 1984). The other abundant species are considered moderately resistant (largemouth bass) or sensitive (rock bass and black crappie, *Pomoxis nigromaculatus*) to acid waters (Magnuson *et al.* 1984; Harvey 1980). The LRL species have very different reproductive strategies and life histories than the more intensively studied salmonids. Besides further defining the absolute and relative sensitivity of these species, in terms of altered production rates and population losses, this study provides an opportunity to evaluate site-specific effect predictions (Brezonik *et al.* 1986) based on various types of data (*i.e.*, laboratory, *in-situ* and survey).

Another major goal of the LRL acidification experiment is to broaden understanding of the direct and indirect mechanisms which control chemical-biological responses to lake acidification. Different mechanisms, such as direct vs food-web mediated, or acid vs dissolved metal determinant, often result in data interpretation discrepancies. Mechanistic studies have played a key role in explaining sensitivities or compensations not discernible from less intensive analysis (Schindler 1987). For example, Mills *et al.* (1987) suggested that, following acidification, the unexpected increase in white sucker (*Catostomus commersoni*) abundance, and unaffected growth and condition were the result of increased invertebrate (dipteran) prey and expanded habitat for fry. Surveys, unless repeated serially, do not reflect temporal trends, and laboratory tests lack the necessary complexity to reveal some mechanisms (Watras and Frost 1989). Analysis of diet, feeding activity, feeding rates and food conversion efficiency have been included in the LRL field program to promote understanding of potentially important indirect mechanisms. Dietary information is correlated with invertebrate prey abundance data obtained by other investigators.

Bioenergetic processes could influence year-class strength, population abundance and production through two mechanisms. One model assumes that direct physiological stress (Selye 1956) results in reduced food conversion efficiency and growth-survival effects on sensitive life stages. The second model assumes changes in prey diversity alter foraging efficiency, resulting in growth-survival and year-class strength changes (Schoener 1969; Werner and Hall 1974). For most species and life stages in LRL, analysis of these effects is based on measurement of diet, growth, and condition. However, field work also includes detailed analyses of feeding attack rates, food consumption rates and food conversion efficiency of Age 0 largemouth bass during the period from early July through late August.

Laboratory toxicological, physiological and histomorphological studies are being conducted in conjunction with field work. There is significant reciprocal cause-effect explanation value associated with coordinating the field and laboratory research. The coordinated effort is useful in separating direct from indirect effects of low pH and elevated aluminum on LRL fish. Aluminum is being studied in conjunction with pH to aid in relating results from LRL to other systems. Aluminum concentrations are not expected to exceed 30 μg/L in LRL throughout the experiment (Pat Brezonik, Univ. of MN, unpublished data), but aluminum has been identified as an important toxic cofactor with low pH in other geographic areas (Baker 1982). The utility of laboratory data for estimating field impacts is also being evaluated. Juvenile stages of the four primary LRL fish species have been exposed to low pH in simulated LRL water to determine relative sensitivity and to observe gill structure and blood osmolality changes (Leino *et al.* 1987; McCormick *et al.* 1987). Additionally, this paper reports effects of low pH and elevated aluminum on the most sensitive embryo and larval life stages of yellow perch, largemouth bass, and rock bass. Research on embryo-larval stages of black crappie will be conducted during 1988. Additional studies relate histological, morphological, and physiological changes occurring in the field to those resulting from laboratory exposure to similar water quality conditions (McCormick *et al.* 1987).

Methods and Materials

The process of designing the experimental work began with project investigators developing a set of system response predictions based on relevant literature and baseline information. These null hypotheses were formally recorded (Brezonik *et al.* 1986) prior to the acidification phase of the study and served as the foundation for field and laboratory research designs.

Field Studies

Reproduction and Early Life History: Reductions only in black crappie year-class strength were predicted to occur in response to the pH 5.6 treatment as a result of reduced survival of fry (Brezonik *et al.* 1986). Field work was designed to measure changes in reproductive activity, embryo survival and fry survival. Following yellow perch spawning in the spring, the number of embryo masses, the average number of embryos per mass, and embryo viability were estimated from shoreline surveys. Data were collected along offshore transects by divers

and along the emergent shore vegetation by observers in boats. Yellow perch embryo density-depth relationships described from the surveys were used to estimate the number of embryo masses spawned in each basin.

Following hatch, larval yellow perch were collected at 3–4 day intervals using a push-net (700μ mesh; 118 × 70 cm opening; Miller 1973). Collections were made near the surface and at 1.2 m. A total of 6 metered pushes were made per day in each basin after sunset. Abundance, growth, food habits, standing crop and production rates were estimated from the catches. Density was multiplied by inhabited water volume, considered to be the basin volume within 2°C of surface temperature (above the thermocline) to estimate total abundance. Comparisons of abundance estimates for the treatment and reference basins were made by 2-way analysis of variance. Instantaneous mortality was estimated from the change in the natural logarithm of abundance over time using linear regression (Ricker 1975).

Reproductive success of the three centrarchid species was defined through nest surveys conducted on alternate days throughout the period of nesting. Location, physical characteristics of each site, indices of male aggressiveness, male size, the occurrence of predation, and fry abundance were recorded for each site. Male size was measured with a calibrated mask-bar (Swenson *et al.* 1988). Nests that produced fry to the swim-up stage were considered successful.

Abundance of fry over largemouth bass nests was estimated from field measurements of the length, width and height of each school, and the mean distance between fry within the school. A mathematical model was used to estimate abundance from the field measurements. The model was calibrated by reconstructing two fry schools in the laboratory from field measurements. These three-dimensional reconstructions consisted of objects placed the mean fry distance apart for two distinctly different school shapes. The model predicted the number of objects present to within 10%.

Both black crappie and rock bass fry disperse from nest sites shortly after hatching. Black crappie fry were captured by push-netting in the spring, coincident to larval yellow perch sampling. Rock bass, however, are difficult to capture because they apparently occupy areas of heavy cover (Powles *et al.* 1980). Sampling of these areas would have been disruptive to other research projects being conducted on LRL. Therefore, active sampling for larval rock bass was not attempted.

Seine, minnow traps, hand pulled trawl, and ⅛ in. frame-nets were used during August to collect samples of Age 0 fish. Age 0 largemouth bass were captured in large enough numbers to allow estimation of abundance by mark-recapture methods. Shoreline snorkeling was used to estimate the proportion of marked (caudal fin clips) and unmarked fish. Age 0 largemouth bass survival was estimated from the difference between spring and fall abundance estimates. Growth of Age 0 largemouth bass was estimated from daily mean weights of fish collected throughout the period after hatching. Instantaneous growth rates were determined by regression analysis. Average size differences between basins were compared using analysis of covariance.

Feeding Bioenergetics: Reductions in feeding attack rates, diet diversity, food conversion efficiency and growth of Age 0 largemouth bass were predicted to occur in response to the pH 5.6 treatment (Brezonick *et al.* 1986). Feeding attack rates of Age 0 largemouth bass were estimated by observers swimming along shorelines. Individual fish randomly selected from small groups were observed for timed periods (usually 60 sec) and the number of feeding attacks was recorded. Attack success, water temperature, time of day, and sky conditions were also recorded. The average number of feeding attacks was calculated for each sampling period to identify temporal changes. Total length of the observed fish was estimated by the mask-bar technique (Swenson *et al.* 1988) and recorded along with the number of fish in each group. Samples were also collected by spear (from observed groups), trawl, seine and minnow trap to obtain length, weight and diet data.

Food consumption rates were estimated by multiplying feeding attack rate by estimated attack success for various attack types (pelagic, surface feeding, benthic or off vegetation) and the percentage occurrence of the corresponding prey type in stomach samples. Representative prey were measured for conversion to dry weight (Bottrell *et al.* 1976; Smock 1980). Estimated biomass of macroinvertebrates consumed was corrected for using prey size-evacuation relationships (Windell 1966). Food conversion efficiency of Age 0 largemouth bass was estimated by dividing dry weight consumed per day by average daily weight gain for the time period. Average daily weight gain was estimated from instantaneous growth rates. Dry weight of Age 0 largemouth bass was estimated to be 20% of wet weight (Steven Poirier, UW-Superior, unpublished data).

Abundance and Population Structure: Changes in larval survival, feeding, food conversion, and growth of young should eventually alter population structure and abundance. Changes in abundance, age structure, survival and recruitment were not predicted to be detectable based on analysis of adult age classes until later in the experiment (Brezonik *et al.* 1986). Abundance and age structure of fish ≥Age 1 was determined from extensive capture-recapture surveys conducted during the spring and fall. Frame nets were used primarily during the spring survey and angling was the primary sampling technique during the fall recapture period. Yellow perch were batch-marked with fin clips. Other species were tagged (Floy FD67D) to permit individual identification, and batch-marked for estimation of tag loss rates. Fish were anesthetized prior to tagging, and total length, weight, and scale samples were collected. Stomachs were pumped to obtain diet information from representative samples. To estimate post-handling mortality, all fish were placed in recovery cages for two hr prior to release. Population size was estimated by Chapman single census and Jolly-Seber multiple recapture estimators (Ricker 1975). Age was estimated from scales for the Centrarchid species and from otoliths of yellow perch. Validation of age assignments based on scales was made by comparison with previous estimates for fish recaptured in successive years.

Laboratory Exposures

Prespawning adult yellow perch and newly fertilized rock bass and largemouth bass embryos were obtained from LRL and transported to the Environmental Research Laboratory-Duluth. Adult yellow perch were held in Lake Superior water (hardness 45 mg/L as $CaCO_3$) until spawning. Embryos taken directly from LRL as well as those spawned in the laboratory were exposed to an array of pH and aluminum concentrations under laboratory conditions.

The exposures were conducted in a continuous flow diluter system modified from Benoit *et al.* (1982) with a 12 hr light, 12 hr

dark photoperiod. The exposure medium was produced by diluting Lake Superior water with deionized Lake Superior water to achieve the following characteristics similar to LRL: calcium, 1.6 mg/L; hardness, 6.5 mg/L $CaCO_3$; and alkalinity, 88.5 μeq/L HCO_3. Reagent grade sulfuric acid was added to provide mean pH levels (± 1 SD) of 6.9 (6.7–7.1), 5.6 (5.4–5.7), 5.0 (4.9–5.1), and 4.5 (4.4–4.6). Free CO_2 remained below 4 mg/L. Aluminum chloride hexahydrate was added to provide nominal inorganic monomeric concentrations (LaZerte 1984) of 0, 25, 50, and 100 μg Al/L. Exposures were conducted in 8 L glass aquaria receiving approximately 75 ml/minute of 20°C (14°C for yellow perch) water to each replicate tank.

Newly fertilized embryos were incubated in 237-ml glass jars with screen bottoms immersed in the exposure tanks. Embryo mortality, hatching success and larval mortality were recorded daily. Survival was measured through seven days after hatching. Results are presented as the cumulative effects on embryo and larval survival. Dry weight of largemouth bass and rock bass fed *ad libitum* was measured 19 and 22 days after hatching, respectively. Survival data were analyzed by arcsin square root transformation and evaluated by analysis of variance. Tukey's test and student's t-test were used to detect significant differences among means (SAS 1985).

Results

Field Studies

Reproduction and Early Life History: Reproductive activity and embryo development of yellow perch were not altered by acidification to pH 5.6. During both 1985 and 1986, the average number of yellow perch embryos per mass was similar between basins (3,300–3,700) and embryo viability remained above 98%. A calculated average of 35 million embryos was produced in the treatment basin, while a calculated average of 20 million was produced in the reference basin.

Analysis of push-net catches showed that abundance estimates of yellow perch fry were higher in the reference basin during 1985 ($p < 0.05$), when average weights were similar between the two basins. Daily abundances were similar in the two basins during 1986. Average weights of larval yellow perch appeared higher in the reference basin during 1986 but differences were not significant. Instantaneous mortality and growth were estimated from the daily abundance and weight estimates using regression analysis (Table 1). Comparison of the mortality and growth rates showed no significant differences between basins (Table 1).

Daily average weights were multiplied by daily abundance estimates to determine mean standing crop. Biological production was estimated by multiplying mean standing crop by instantaneous growth rates (Ricker 1975). The standing crop and biological production estimates were similar within each basin during 1985 and 1986. Although not significantly different, the mean standing crop and biological production estimates were approximately 25% lower in the treatment basin during the pH 5.6 treatment (Table 1).

All Centrarchid species nested during the pH 5.6 treatment. The number of nests observed for all species was greater in the treatment basin during both years of the pH 5.6 treatment, reflecting higher adult abundances (Table 2). Following black crappie nesting, fry were abundant in subsequent treatment basin push-net samples. A total of 66 and 138 fry were captured from the treatment basin during 1985 and 1986, respectively. Only one black crappie fry was collected from the reference basin, where no nests were observed either year.

No largemouth bass nests were successful in the reference basin during 1985. Seven hundred fry were transplanted from the treatment basin in an effort to establish a 1985 year-class. Treatment basin largemouth bass produced an estimated 54,600 fry in 1985. During 1986, the 24 successful treatment basin nests produced an estimated 65,800 fry, while 13,200 were produced from 4 successful reference basin nests. Estimated fall abundances of Age 0 largemouth bass in the treatment and reference basins were 4000 and 2700, respectively during 1986. Comparison of spring and fall estimates suggests both density and mortality were higher in the treatment basin.

Average weight of fry collected from nest sites appeared to increase faster in the reference basin (Figure 1) but differences were not significant. To eliminate bias associated with more extensive spawning of the larger treatment basin population, growth estimates for the period following dispersal from nests were based only on mean weights of Age 0 largemouth bass from early nests in both basins. Length-frequency analysis was used to identify young from early nests. Instantaneous growth rates calculated for this period were similar, but average weight of Age 0 largemouth bass in the reference basin was greater ($p < 0.01$). The higher elevation of the reference basin growth relationship demonstrates that average weight and weight gain per unit time were significantly greater in the reference basin. Differences in temperature during nesting and growth periods were not significant between the basins, based on two-way analysis of variance. Date of embryo hatch was also similar between basins; fry production peaked on the same date.

Rock bass nesting occurred in both basins during the pH 5.6 treatment (Table 2). However, fry were

Table 1. Larval yellow perch mortality, growth, standing crop, and production in the treatment (T) and reference (R) basins of Little Rock Lake

Year	Basin	Instantaneous daily rates (±2 SE)		Standing crop[a] (g/basin)	Production rate[a] (g/basin/day)
		Mortality	Growth		
1985	T	0.1415 ± .0282	0.1046 ± .1038	1,076	113
1985	R	0.1865 ± .0372	0.0949 ± .0102	1,429	137
1986	T	0.1665 ± .0878	0.1109 ± .0187	1,012	112
1986	R	0.1914 ± .0903	0.1084 ± .0084	1,511	164

[a] Based on average inhabited water volumes of 2.606×10^6 m^3 and 2.216×10^6 m^3 in the treatment and reference basins, respectively

Table 2. Abundance (±95% C.I.) and number of nests of the four dominant species in the treatment (T) and reference (R) basins of Little Rock Lake, with a list of other species

Species	Basin	Abundance		Total nests (number successful)	
		1985	1986	1985	1986
Largemouth bass	T	423 ± 134	400 ± 129	40 (17)	90 (24)
	R	120 ± 32	103 ± 40	3 (0)	36 (4)
Rock bass	T	690 ± 64	513 ± 160	19 (0)	16 (0)
	R	308 ± 49	203 ± 102	3 (0)	14 (0)
Black Crappie	T	104 ± 58	78 ± 43	2 (0)[a]	20 (8)
	R	25 ± 41	no estimate	0 (0)[a]	0 (0)
Yellow perch	T	25,326 ± 1,793	30,643 ± 3,760		
	R	16,728 ± 1,342	27,068 ± 4,432		
Less Abundant Species					
Mudminnow (*Umbra limi*)					
Golden shiner (*Notemigonus crysoleucus*)					
Hornyhead chub (*Nocomis biguttatus*)					
Creek chub (*Semotilus atromaculatus*)					
White sucker (*Catostomus commersoni*)					
Common shiner (*Notropis cornutus*)					
Walleye (*Stizostedion vitreum*)					

[a] No successful nests were observed but fry were subsequently captured in pushnet samples

not observed in or near nests in either basin. No individuals representing the 1985–1986 year-classes were captured in the treatment basin during 1985–1987. A total of 5 Age 0 and Age 1 rock bass representing the 1985 and 1986 year-classes were captured from the reference basin during the same period. Over 100 Age 1 rock bass representing the 1983 year-class were captured during 1984 using similar sampling procedures.

Feeding Energetics: Feeding attack rates of Age 0 largemouth bass were similar between basins and years. Diet analysis showed *Polyphemus* spp. was the predominant crustacean consumed from 1984–1986. Comparison of prey in stomachs suggests no major shifts occurred in diet or in the size of prey consumed. Consumption of dipterans increased in both basins during 1986. Estimated food consumption was higher during August but generally similar between the two basins (Table 3). However, weight gain calculated from the relationships presented in Figure 1, and estimated food conversion efficiency were higher in the reference basin (Table 3).

Abundance and Population Structure: Extensive tagging efforts began in the fall 1984. Preliminary estimates based on Chapman's single census estimator suggested that abundance of most species was higher in the treatment basin (Table 2). Comparison of estimates from 1984 to 1986 indicate no changes have occurred.

Laboratory Exposures

Larval yellow perch survival declined to 25 percent at pH 4.5, but still was not significantly different

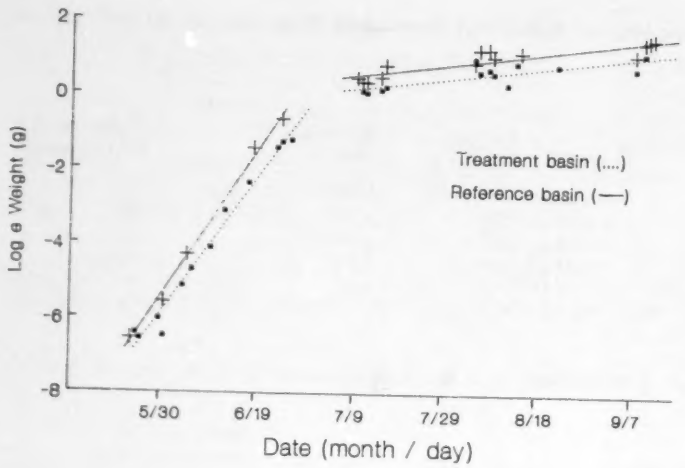

Fig. 1. Growth of Age 0 largemouth bass in the treatment (■) and reference (+) basins of Little Rock Lake during 1986. Earlier weights are fry from nests in the treatment (Y = −32.46 + 0.178X) and reference (Y = −34.19 + 0.192X) basins. Later weights are for fingerlings in the treatment (Y = −2.63 + 0.015X) and reference (Y = −2.62 + 0.016X) basins

Table 3. Food consumption, growth and conversion efficiency of Age 0 largemouth bass in the treatment (T) and reference (R) basins of Little Rock Lake during 1986

	July		August	
	T	R	T	R
Food consumption (mg/day)	14.2	17.6	61.0	48.5
Growth (mg/day)				
Wet weight	17.4	27.0	31.1	47.7
Dry weight	3.5	5.4	6.2	9.5
Conversion efficiency	0.25	0.31	0.10	0.20

from the pH 6.9 control. At pH 5.0 with 100 μg/L aluminum, survival significantly declined to 5% compared to greater than 50% survival in the other pH 5.0 treatments ($p < 0.05$). Addition of 100 μg/L aluminum at pH 4.5 resulted in total mortality (Figure 2). Larval rock bass survival declined to 46% at pH 5.6, and was significantly reduced at pH 5.0 and below ($p < 0.05$). No significant differences were found between the aluminum treatments at pH 5.0; however, addition of 25 μg/L aluminum increased survival to 23% compared to 7% in the pH 5.0 treatment without aluminum (Figure 2). No larvae survived at pH 4.5. Survival of largemouth bass larvae ranged from 38 to 29% through pH 5.0, while all larvae died after exposure to pH 4.5. Low control survival resulted from a high percentage of egg infertility. Addition of 50 and 100 μg/L aluminum at pH 4.5 increased survival to 8 and 4%, respectively. No differences in growth of larval largemouth bass or rock bass were observed at pH 5.6 or higher.

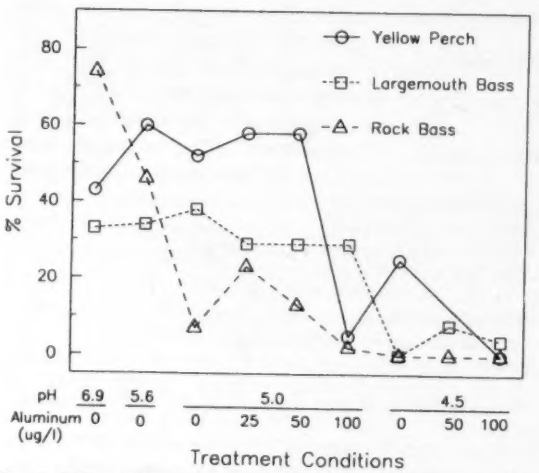

Fig. 2. Effects of lowered pH and elevated aluminum on yellow perch, largemouth bass, and rock bass embryo-larval survival through seven days post-hatching

Discussion

The field work on LRL suggests that the reduction of lake pH to 5.6 decreased recruitment of rock bass. Although survival differences observed in the laboratory tests were not significant, mean survival was considerably lower at pH 5.6 compared to pH 6.9. Rock bass have been reported in lakes below pH 5.0 (Ryan and Harvey 1977), and Weiner and Eilers (1987) have reported pH 5.2 as the minimum for the occurrence of rock bass in Wisconsin lakes. The relatively small catches of 1985–1986 year-class rock bass from the LRL reference basin compared with the much larger catches of the 1983

year-class, suggest failure to catch rock bass in the treatment basin may be due in part to the occurrence of weak year-classes. Verification of recruitment failure is dependent upon continued sampling. The importance of successive years of data in defining fish population responses has been demonstrated by nine years of research on the acidified Lake 223 (Mills *et al.* 1987). Year-class strength in 7 additional area lakes is being monitored by cooperating scientists (Magnuson 1982) and will be compared with LRL in assessing cause-effect relationships. *In-situ* experiments on early life stages of rock bass will be conducted in LRL during 1988 to further compare field and laboratory responses. Additional stresses under field conditions (*e.g.*, food availability, predation, heavy metals) could result in higher mortality in the field. Hutchinson and Sprague (1986) found low pH, aluminum, zinc, and copper to be additive in their toxicity to fish in soft water.

Higher density of Age 0 largemouth bass in the treatment basin during 1986 was considered as a possible cause for slower growth through density-dependent food competition. However, food consumption by treatment basin fish was estimated to be higher per unit weight, suggesting food competition was not a limiting factor. Therefore, food conversion efficiency rather than food availability may have resulted in significantly lower size. The method used to estimate food consumption is still being refined and the results must be regarded as preliminary. However, if substantiated by further analysis, sublethal stress appears to be the responsible mechanism. Shuter *et al.* (1980) showed that the size attained by fall was critical to Age 0 smallmouth bass (*Micropterus dolomieui*) survival and recruitment. Laboratory growth studies conducted with the same size classes of LRL largemouth bass, using *ad libitum* feeding, failed to detect significant growth differences at pH 4.5 (J. H. McCormick, unpublished data). Histomorphological analyses of gill tissues from laboratory and field exposed largemouth bass may be useful in substantiating the occurrence of stress.

Changes in prey community structure were expected to influence feeding, conversion efficiency and growth (Magnuson *et al.* 1984; Mills *et al.* 1987). Optimal foraging theory predicts that a shift from larger to smaller prey species can result in greater energy expenditure per unit volume ingested by predators, reducing conversion efficiency and growth (Schoener 1969; Werner and Hall 1974). Mills *et al.* (1987) found reductions in prey size and biomass resulted in reduced growth and condition of lake trout (*Salvelinus namaycush*) in Lake 223. The relative abundance of potential invertebrate fish-food organisms has changed in the treatment basin (T. Frost, personal communication) and subsequent diet analyses of LRL fish may reveal relationships to feeding and growth. However, failure to detect such a change could be due, in part, to limitations in the frame of reference for this phase of the work. To improve it in relation to understanding feeding energetics, sampling of additional reference lakes was initiated during 1987. Two lakes located within 40 km of LRL were selected, both with largemouth bass populations representing the top predator. Dorothy-Dunn lake includes a more diverse prey species complex and a higher pH (7.8). The second lake, Camp Lake, has a pH similar to the LRL reference basin with a prey species complex dominated by bluegill sunfish (*Lepomis macrochirus*).

Field studies show yellow perch persist at low pH (Magnuson *et al.* 1984; Weiner and Eilers 1987) and possible between-basin differences in estimated fry standing crop and production rates observed in LRL may be the result of sample variability. Identifying indirect mechanisms represents one approach useful in assessing reliability of this type of result. Diet, changes in the zooplankton community and temperature data will be analyzed to determine whether differences in fry abundance, standing crop and production rates can be traced to some acid-related change in prey availability or other variable. These analyses may indicate that production of even "resistant" yellow perch is influenced in some lakes by acidification through indirect mechanisms. Forney (1977) has shown that changes in larval yellow perch survival do influence year-class strength and abundance. Continued sampling of adult populations will determine if differences in year-class strength occurred during the pH 5.6 treatment.

Some conclusions are possible from examination of the laboratory results alone. As observed for other species (*e.g.*, brook trout, *Salvelinus fontinalis;* Baker and Scholfield 1982) aluminum appears to be protective for rock bass and largemouth bass but not for yellow perch. Aluminum, in combination with low pH, began to exert a toxic effect at concentrations between 50 and 100 μg/L. The relative sensitivity of fish species to aluminum differs from their sensitivity to pH. Yellow perch suffered nearly 100% mortality at pH 5.0 and 100 μg/L aluminum, while largemouth bass were unaffected. At pH 4.5 without aluminum, largemouth bass were more sensitive. It is also noteworthy that growth differences of post-fry stages were not observed in laboratory treatments through pH 5.6. No noticeable changes were observed in the ionoregulatory capacity of fish sampled from LRL or in laboratory

fish exposed to pH 5.6 and 6.1 (McCormick *et al.* 1987).

In summary, the laboratory results indicate that species, lifestage, and water quality constituents can influence acid impact on fisheries. The preliminary field results on LRL complement the findings of Mills *et al.* (1987) on Lake 223 and suggest that indirect mechanisms may be important during the early stages of acidification.

References

Baker JP (1982) Effects on fish of metals associated with acidification. In: Johnson RE (ed) Acid rain/fisheries. Am Fish Soc, Bethesda, Maryland, pp 165–176

Baker JP, Scholfield CL (1982) Aluminum toxicity to fish in acidic water. Water Air Soil Pollut 18:289–309

Benoit DA, Mattson VR, Olson DL (1982) A continuous-flow minidiluter system for toxicity testing. Water Res 16:457–464

Bottrell HH, Duncan A, Gliwicz ZM, Grygierek E, Herzig A, Hillbricht-Ilkowska A, Kurasawa H, Larsson P, Weglenska T (1976) A review of some problems in zooplankton production studies. Norw J Zool 24:419–456

Brezonik PL, Baker LA, Detenbech N, Eaton JG, Frost TM, Kratz TK, Magnuson JJ, McCormick JH, Perry JE, Rose WJ, Shepard BK, Swenson WA, Watras CJ, Webster KE (1986) Experimental acidification of Little Rock Lake, Wisconsin: Baseline studies and predictions of lake responses to acidification. Special Report 7. Water Resource Research Center, University of Minnesota, St. Paul, Minnesota, 43 pp

Forney JL (1977) Reconstruction of yellow perch (*Perca flavescens*) cohorts from examination of walleye (*Stizostedion vitreum*) stomachs. J Fish Res Board Can 34:925–932

Harvey HH (1980) Wide spread and diverse changes in the biota of North American lakes and rivers coincident with acidification. In: Drablos D, Tollan A (eds) Ecological Impact of Acid Precipitation. Proc Int Conf, Sandefjord, Norway. SNSF Project, Oslo, Norway, pp 93–97

Hutchinson NJ, Sprague JB (1986) Toxicity of trace metal mixtures to American flagfish (*Jordanella floridae*) in soft, acidic water and implications for cultural acidification. Can J Fish Aquat Sci 43:647–655

LaZerte BD (1984) Forms of aqueous aluminum in acidified catchments of central Ontario: A methodological analysis. Can J Fish Aquat Sci 41:766–776

Leino RL, McCormick JH, Jensen KM (1987) Changes in the gill histology of fathead minnows and yellow perch transferred to soft water or acidified soft water with particular reference to chloride cells. Cell Tissue Res 250:389–399

Magnuson JJ (1982) Comparative studies of a suite of lakes in Wisconsin. Long-term ecological studies progress report to the National Science Foundation

Magnuson JJ, Baker JP, Rahel F (1984) A critical assessment of the effects of acidification on fisheries. Phil Trans Royal Soc London B305:501–506

McCormick JH, Jensen KM, Leino RL, Stokes GN (1987) Fish blood osmolality, gill histology, and oocyte atresia as early warning acid stress indicators. Annals Soc r zool Belg 117 Supplement 1:309–319

Miller JM (1973) Quantitative push-net system for transect studies of fish and macrozooplankton. Limnol Oceanogr 18:175–178

Mills KM, Chalanchuk SM, Mohr LC, Davies IJ (1987) Response of fish populations in Lake 223 to 8 years of experimental acidification. Can J Fish Aquatic Sci 44 Supplement 1:114–125

Powles PM, Vandeloo DR, Clancy B (1980) Some features of larval rock bass, *Ambloplites rupestris* (Rafinesque) development in central Ontario. Proc Fourth Annl Larval Fish Conf

Ricker WE (1975) Computation and interpretation of biological statistics of fish populations. Fish Res Board Can Bull 191, 382 pp

Ryan PM, Harvey HH (1977) Growth of rock bass, *Ambloplites rupestris*, in relation to the morphoedaphic index as an indicator of an environmental stress. J Fish Res Board Can 34:2079–2088

SAS User's Guide (1985) SAS Institute Inc, Cary, North Carolina

Schindler DE (1987) Detecting ecosystem response to anthropogenic stress. Can J Fish Aquat Sci 44 Supplement 1:6–25

Schoener TW (1969) Models of optimal size for solitary predators. Am Nat 103:277–313

Selye H (1956) The stress of life. McGraw Hill, New York, 324 pp

Shuter BJ, McLean JA, Fry EFJ, Regier HA (1980) Sochastic simulation of temperature effects on first-year survival of smallmouth bass. Trans Am Fish Soc 109:1–34

Smock LA (1980) Relationship between body size and biomass of aquatic insects. Freshwater Biol 10:375–383

Swenson WA, Gobin WP, Simonson TD (1988) A calibrated mask-bar for underwater measurement of fish. N Am J Fish Manage 8:(In press)

Watras CJ, Frost TM (1989) Little Rock Lake: Perspectives on an experimental, ecosystem approach to seepage lake acidification. Arch Environ Contam Toxicol 18:157–165

Weiner JG, Eilers JM (1987) Chemical and biological status of lakes and streams in the upper midwest: Assessment of acidic deposition effects. Lakes and Reservoir Manage 3:365–378

Werner EE, Hall DJ (1974) Optimal foraging and the size selection of prey by the bluegill sunfish (*Lepomis macrochirus*). Ecology 55:1042–1052

Windell JT (1966) Rates of digestion in the bluegill sunfish. Invest Indiana Lakes and Streams 7:185–214

Manuscript received March 19, 1988.